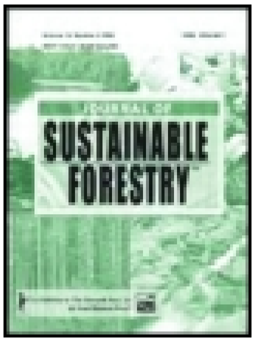

Journal of Sustainable Forestry

ISSN: 1054-9811 (Print) 1540-756X (Online) Journal homepage: http://www.tandfonline.com/loi/wjsf20

Postsettlement Changes in Natural Fire Regimes and Forest Structure

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To cite this article: W. W. Covington & M. M. Moore (1994) Postsettlement Changes in Natural Fire Regimes and Forest Structure, Journal of Sustainable Forestry, 2:1-2, 153-181, DOI: 10.1300/J091v02n01_07

To link to this article: https://doi.org/10.1300/J091v02n01_07

Published online: 18 Oct 2008.

ECOLOGICAL AND HISTORICAL PERSPECTIVES

Postsettlement Changes in Natural Fire Regimes and Forest Structure: Ecological Restoration of Old-Growth Ponderosa Pine Forests

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The authors thank Robert W. Mutch, Lyn Morelan, Penny Morgan, and Maia Enzer for many useful comments and suggestions for improving this manuscript.

This research was supported by research agreements including Northern Arizona University (Organized Research Program and Bureau of Forestry Research), the McIntire-Stennis Cooperative State Research Service, and the Salt River Project. In kind support was provided by Kaibab Renewable Resources.

An early unrefereed version (Covington and Moore 1992) of this paper was published in the proceedings of a workshop on old-growth forests held in Portal, Arizona, March 9-13, 1992.

[Haworth co-indexing entry note]: "Postsettlement Changes in Natural Fire Regimes and Forest Structure: Ecological Restoration of Old-Growth Ponderosa Pine Forests." Covington, W. W. and M. M. Moore. Co-published simultaneously in the *Journal of Sustainable Forestry* (The Haworth Press, Inc.) Vol. 2, No. 1/2, 1994, pp. 153-181; and: *Assessing Forest Ecosystem Health in the Inland West* (eds: R. Neil Sampson and David L. Adams) The Haworth Press, Inc., 1994, pp. 153-181. Multiple copies of this article/chapter may be purchased from The Haworth Document Delivery Center [1-800-3-HAWORTH; 9:00 a.m. - 5:00 p.m. (EST)].

ABSTRACT. Heavy livestock grazing, logging, and fire exclusion associated with Euro-American settlement has brought about substantial changes in forest conditions in western forests. Thus, old-growth definitions based on current forest conditions may not be compatible with the natural conditions prevalent throughout the evolutionary history of western forest types. Detailed analysis of data from two study areas in the southwestern ponderosa pine type suggests that average tree densities have increased from as few as 23 trees per acre in presettlement times to as many as 851 trees per acre today. Associated with these increases in tree density are increases in canopy closure, vertical fuel continuity, and surface fuel loadings resulting in fire hazards over large areas never reached before settlement. In addition, fire exclusion and increased tree density has likely decreased tree vigor (increasing mortality from disease, insect, drought, etc.), herbaceous and shrub production, aesthetic values, water availability and runoff, and nutrient availability, and also changed soil characteristics and altered wildlife habitat. To remedy these problems and restore these forest ecosystems to more nearly natural conditions, and maintain a viable cohort of old age-class trees, it will be necessary to thin out most of the postsettlement trees, manually remove heavy fuels from the base of large, old trees, and reintroduce periodic burning.

Between the two extremes of passively following nature on the one hand, and open revolt against her on the other, is a wide area for applying the basic philosophy of working in harmony with natural tendencies. (H. J. Lutz 1959)

INTRODUCTION

Understanding natural ecological conditions and processes, before significant impact by Euro-American settlement (Kilgore 1985), is central to developing ecologically coherent forest management programs (e.g., Vogl 1974; Franklin 1978; Harris 1984; Kilgore 1985; Forman and Godron 1986). This is particularly true for management of landscape diversity with remnant natural patches of old-growth forests (Bonnicksen and Stone 1985; Parsons et al. 1986; Moir and Dieterich 1988; Forman and Godron 1986; Booth 1991) or managing for the development of old-growth and other stages of forest development (Thomas 1979; Hoover and Wills 1984; Moir and Dieterich 1988). However, heavy livestock grazing, logging, and fire suppression associated with Euro-American settlement have brought about substantial changes in forest conditions in western forests, so much so that current conditions may be decidedly "unnatural." In particular, the exclusion of natural fires has led to increased tree densities and associated shifts in ecosystem structure, fire hazard, disturbance regimes, and wildlife habitat in some western forest types. Thus, old-growth definitions based on current forest structure may not be compatible with the natural conditions prevalent throughout the evolutionary history of the organisms living in western forests. For these reasons, we believe that planning and management for old-growth within ponderosa pine and other forest types must include an understanding of past (presettlement), present, and future conditions. Failing to do so may lead to less than desirable forest conditions in the future.

In this paper we present a general discussion of changes in natural fire regimes, then examine in more detail the evidence for such changes in the southwestern ponderosa pine type and how these changes have affected overall ecological conditions. Finally, we close with a brief discussion of possible methods for remedying some of the problems associated with postsettlement changes in western forests.

NATURAL FIRE REGIMES

Understanding the natural disturbance regimes under which a particular species evolved is central to predicting the ecological consequences of management activities (e.g., Bormann 1981; White 1979). Periodic wildland fire has played a central role in the evolution of forest and woodland ecosystems throughout the western United States (Parsons 1981; Kilgore 1981; Covington et al. 1994b). In fact, many species and forest types worldwide appear to be dependent upon a particular frequency and intensity of fire for their survival (Mooney 1981; Parsons 1981).

Fire regimes have been classified according to frequency, intensity, size, and type (Heinselman 1981; Kilgore 1981; Sando 1978). Frequency, or burning interval, has been defined as the average return period for fire burning through a particular vegetation type. Sando defined frequent fires as fires which occur at intervals of 1-10 years; infrequent fires by his definition are fires occurring at intervals greater than 10 years, often as infrequently as once every 20-300 years. Kilgore (1981) separated frequent from infrequent fires at 25 years.

Although fire intensity has been used as a qualitative term (e.g., light surface fire vs. severe crown fire), some authors argued for a quantitative definition such as fire line intensity (Sando 1978; Kilgore 1981). Specifically, they recommended Byram's fire line intensity, a product of heat yield per unit area (BTU per square foot) and the rate of fire spread (feet per second). The resulting units are BTUs per foot per second. To avoid confusing qualitative and quantitative definitions, we recommend that qualitative differences be referred to as fire severity, and that the term fire intensity be reserved for a more quantitative measure.

Fire regimes have also been characterized according to size (Table 1). However, no generally agreed upon classification exists; certainly other size classes may be more appropriate for specific applications.

Fire type has been classified into as few as two (surface vs. crown) to six or more categories. For example, Heinselman (1981) differentiated fire type into light surface fires, severe surface fires, crown fires, and combinations of the three. Kilgore (1981) used the terms low intensity surface fire, high intensity surface fire, stand replacement fire, and combinations of these three categories. Several authors have related fire severity to fire intensity (Byram 1959; Van Wagner 1973; Albini 1976; Sando 1978). Integrating these views on intensity: severity relationships, Sando (1978) concluded that at low to moderate (0-1200 BTU/ft/sec) fire line intensity, complete mortality of overstory vegetation would not occur. At levels above 1200 BTU/ft/sec, nearly complete overstory mortality would be expected.

For this discussion we will use Sando's (1978) classification of natural fire regimes. However, more comprehensive classification strategies should be developed for designing adaptive management experiments.

TABLE 1. Examples of fire size classification.

Fire size (acres)	USDA Forest Service Wildfire Classes	Heinselman (1981) Classes
<0.25	A	small
0.26-9	B	small
10-99	C	small
100-299	D	medium
300-999	E	medium
1,000-4,999	F	large
5,000-9,999	G	large
>10,000	G	very large

Nonetheless, for brevity's sake we will use the following four types in this paper:

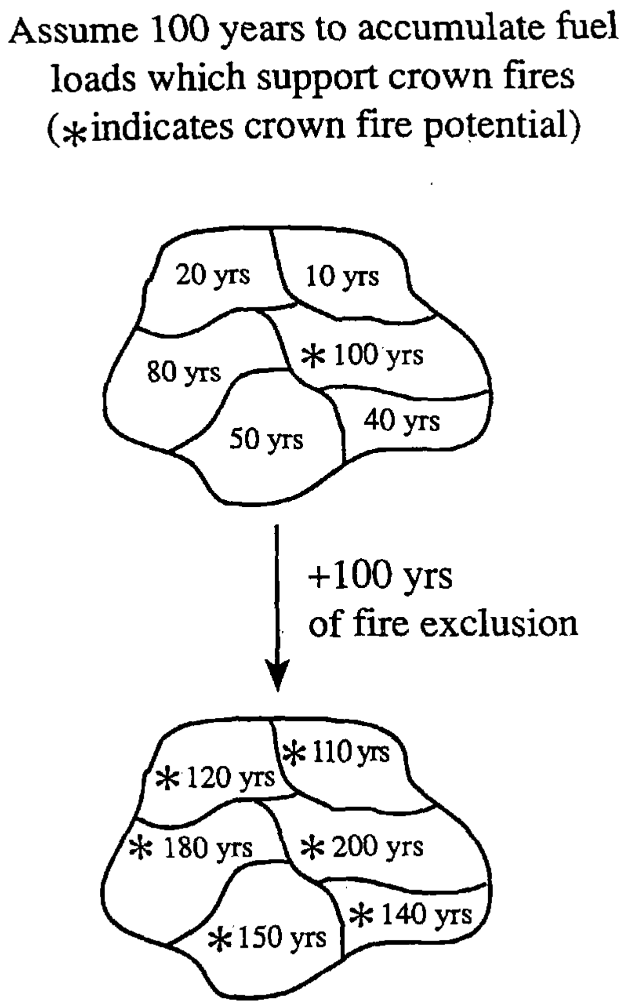
Type one: frequent fires (1-25 yr) of low to moderate intensity (< 1200 BTU/ft/sec) (e.g., ponderosa pine, lower elevation [warmer and drier sites] mixed conifer, giant sequoia, southern pine forests, short grass and mixed grass prairies, savannahs).
Type two: infrequent fires (> 25 yr) of high intensity (> 1200 BTU/ft/sec)(e.g., boreal and subalpine spruce-fir, higher elevation [cooler and wetter] mixed conifer forests, temperate rain forests).
Type three: frequent fires of high intensity (e.g., tall grass prairie).
Type four: infrequent fires of low to moderate intensity (e.g., deserts, tundra, mesic deciduous forests).

Fire exclusion affects each of these fire regimes differently. Although it typically causes major shifts in ecosystem structure and function (see below), from a fire control perspective the central concern is when shifts occur between fire regimes. Only small remnants of type three exist, virtually all of it having been converted to agriculture. However, under natural conditions, fire suppression would not have been practical in this type because of its extreme fire behavior. Fire exclusion in type four has little effect on the fire regime because excess flammable organic matter rarely accumulates.

Over time, fire exclusion in type two will result in rescaling of fire size as more patches reach a condition which will support crown fire. For example, if we assume that 100 years are necessary for a particular vegetation type to accumulate sufficient fuel to support a crown fire, then after 100 years of successful fire exclusion, all of the area in that type would be capable of supporting crown fire (Figure 1). Thus, in type two, fires become larger over time.

The greatest change in fire regimes following fire exclusion is in type one. Here the natural fire regime was frequent enough to keep surface fuel loads low and to thin out trees so that canopy fuels were separated both vertically and horizontally. With fire exclusion in this type, surface fuels accumulate and trees become established gradually, providing a fuel ladder and increasing canopy closure. These changes in fuel structure lead to a shift from light surface fires to intense stand replacement crown fires characteristic of the type two fire regime. Continued fire suppression in this type might well lead to the same changes seen in type two fire regimes, i.e., the coalescing of patches into larger and larger areas capable of supporting crown fire (Figure 1). The classic example of a type one natural

FIGURE 1. Effects of fire exclusion in landscapes with an infrequent, high intensity natural fire regime (Type two).



fire regime is ponderosa pine ecosystems (Biswell 1972; Kilgore 1981; Weaver 1974).

Now we will turn to a more detailed discussion of ecological consequences of shifts in fire regimes by examining postsettlement changes in southwestern ponderosa pine ecosystems.

POSTSETTLEMENT CHANGES IN SOUTHWESTERN PONDEROSA PINE FORESTS

It is widely acknowledged that fire exclusion and other factors associated with European settlement have greatly altered forest conditions in southwestern ponderosa pine (Cooper 1960; Weaver 1951; Covington and Sackett 1984; White 1985; Covington and Sackett 1986; Covington and Moore 1994a). However, postsettlement changes in the ponderosa pine type are not unique to the Southwest. In fact, studies in Utah (Madany and West 1983; Stein 1987), Montana (Gruell et al. 1982), Idaho (Barrett 1988; Steele et al. 1986), Washington (Weaver 1959), and California (Laudenslayer et al. 1989) suggested that increased tree density, fuel loading, and crown fire occurrence are common consequences of fire exclusion throughout the ponderosa pine type (Kilgore 1981). Simulation studies (van Wagtendonk 1985; Keane et al. 1990) indicated that this phenomenon occurs not only in the pure ponderosa pine type, but also in ponderosa pine/Douglas-fir and mixed conifer forests as well. Thus, although many questions remain regarding the ecological and multiresource implications of postsettlement changes in ponderosa pine forests, there is a wide consensus that today's forests are radically different from those present before European settlement.

Reports from early travelers illustrate the changes in appearance of the ponderosa pine forest since settlement. E. F. Beale's 1858 report is quoted by C. F. Cooper (1960) as follows:

> We came to a glorious forest of lofty pines, through which we have travelled ten miles. The country was beautifully undulating, and although we usually associate the idea of barrenness with the pine regions, it was not so in this instance; every foot being covered with the finest grass, and beautiful broad grassy vales extending in every direction. The forest was perfectly open and unencumbered with brush wood, so that the travelling was excellent. (Beale 1858)

Cooper (1960) stated "The overwhelming impression one gets from the older Indians and white pioneers of the Arizona pine forest is that the entire forest was once much more open and park-like than it is today."

Before European settlement of northern Arizona in the 1860s and 70s, periodic natural surface fires occurred in ponderosa pine forests at frequent intervals, perhaps every 2-12 years (Weaver 1951; Cooper 1960; Dieterich 1980). Extensive study of fire scars suggests that the natural fire size was approximately 3,000 acres (Swetnam and Dieterich 1985; Swetnam 1990).

Several factors associated with European settlement caused a reduction in natural fire frequency and size. Roads and trails broke up fuel continuity. Domestic livestock grazing, especially overgrazing and trampling by cattle and sheep in the 1880s and 1890s, greatly reduced herbaceous fuels. Active fire suppression, as early as 1908 in the Flagstaff area, was a principal duty of early foresters in the Southwest. A direct result of interrupting and suppressing these naturally occurring, periodic fires has been the development of overstocked forests.

Changes in the forest structure (e.g., tree density, cover, and age distributions) in southwestern ponderosa pine forests since European settlement have been blamed for many forest management problems (Biswell 1972; Cooper 1960; Weaver 1974; Covington and Sackett 1990; Covington and Moore 1994a). Forest management problems attributed to fire exclusion and resulting increased tree density in southwestern ponderosa pine include:

1. overstocked sapling patches;
2. reduced tree growth;
3. stagnated nutrient cycles;
4. increased disease, insect infestation, and parasites (e.g., root rot, bark beetle, dwarf mistletoe);
5. decreased forage quality and quantity;
6. increased fuel loading;
7. increased vertical fuel continuity due to dense sapling patches;
8. increased severity and destructive potential of wildfires;
9. increased tree canopy closure;
10. decreased on-site water availability;
11. decreased stream-flow and ground water recharge;
12. shifts in habitat quality for biota; and
13. decreased diversity of native flora and associated food webs.

Evidence for the shift from a type one to a type two fire regime in ponderosa pine since settlement comes from a study by Barrows (1978), later updated by Swetnam (1990). Using USDA Forest Service wildfire statistics they determined that lightning-caused crown fires had increased from 10,127 acres per year in the 1940s to 15,117 acres per year in the 1980s. In the 1970s an average of 33,801 acres per year were burned by wildfire in the Southwest. Both Barrows (1978) and Swetnam (1990) observed that lightning-caused wildfires in the Southwest are getting larger and larger over time, with some fires reaching 10,000-20,000 acres, in contrast to the 3,000 acre surface fires of presettlement times (Swetnam and Dieterich 1985; Swetnam 1990). This represents a three- to six-fold increase in average fire size. Thus, we may be witnessing in the ponderosa pine type the kind of shift in size observed in the type two fire regimes, i.e., the coalescing of patches into larger and larger areas capable of supporting very large (> 10,000 acres) crown fires.

There is little quantitative information on conditions of presettlement forests and woodlands of the Inland West (Covington et al. 1994b). The major writings and research in the southwestern ponderosa pine type deal only with tree densities. As mentioned earlier, Cooper (1960) cited the writings of early expedition leaders, Whipple (1856) and Beale (1858). They reported that the condition of the southwestern ponderosa pine forest " . . . was open and park-like with a dense grass cover." These early descriptions of the open nature of presettlement ponderosa pine forests are in agreement with results of recent research which found that canopy coverage by trees of presettlement origin range from 17% (Covington and Sackett 1986) to 22% (White 1985) of the surface area for unharvested sites near Flagstaff, AZ. In addition, Pearson (1923) noted that "rarely does [ponderosa pine] crown cover reach more than 30% and usually not over 25%."

Cooper (1960) stated that the structure of the southwestern ponderosa pine type in the White Mountains of east-central Arizona is actually that of an all-aged forest composed of even-aged groups. He noted great variation in diameter within a single age class. Using contiguous quadrat analysis (Grieg-Smith 1952) in two stands, Cooper determined that the presettlement trees aggregated into areas ranging from 0.16 to 0.32 acres. White (1985), in a study conducted on the Pearson Natural Area near Flagstaff, noted that successful establishment of ponderosa pine in presettlement times was infrequent (as much as four decades between regeneration events). White also quantified the strong aggregation of ponderosa pine. Using the nearest neighbor method (Clark and Evans 1954), White demonstrated that the aggregation ranged from 3 to 44 stems within a group, with a group occupying an area that ranged from 0.05-0.70 acres. "Ages of stems within a group were also variable with the most homogeneous group having a range of 33 years and the least having a range of 268 years (White 1985)." White's findings of a pattern of uneven-aged groups near Flagstaff are in contrast to the results of Cooper (1960) for the White Mountains. However, Cooper's subsequent observations have lead him to conclude that even-agedness in presettlement ponderosa forests was rare (C.F. Cooper, San Diego State University, San Diego, CA, personal communication 1992). For the southwestern ponderosa pine type, therefore, the data suggest that, at the group-level, the trees are basically all-aged and have sporadic regeneration events (White 1985; Covington and Moore 1994a). At the landscape and regional levels (square miles and larger in size), however, several studies have shown simultaneous regeneration events that were correlated with simultaneous surface fires, and favorable climatic oscillations (e.g., La Niña [opposite pattern of El Niño], Kerr 1988; Swetnam 1990; Savage 1989).

Madany and West (1983) discussed the effects that many years of heavy grazing and fire suppression have had on ponderosa pine regeneration in southern Utah (Zion National Park). They suggested that ponderosa pine seedling survival was probably greater in the early 1900s than in the presettlement days due to reduced competition of grasses (through grazing) with pine seedlings, and the reduced thinning effect that fires once had on seedlings in presettlement times.

Moir and Dieterich (1988) pointed out the importance of understanding the role of the natural, presettlement fire regime in directing successional processes toward ponderosa pine old-growth development and in keeping fuel loading low enough for large trees to survive wildfires. They stated that most of the old-growth in southwestern ponderosa pine forests has deteriorated because recurrent natural fires have been suppressed. Finally, they present an eleven stage model of succession from open meadow through sapling, pole, yellow pine, and dead snag dominated landscape units. Although they do not describe the scale of these units, Cooper's (1960) and White's (1985) results indicate that these units were on the order of a few tenths of an acre.

DETAILED ANALYSIS OF TWO STUDY AREAS IN ARIZONA PONDEROSA PINE

To better understand postsettlement changes in southwestern ponderosa pine, we studied changes in forest conditions for two areas in northern Arizona. One, the Bar-M study area, has soils of volcanic origin and is on the Mormon Lake Ranger District of the Coconino National Forest. The other has soils of limestone origin and is on the North Kaibab Ranger District of the Kaibab National Forest. More detailed descriptions of the study area and methods used are available in Covington and Moore (1994a) and Covington and Moore (in preparation).

Study Areas

The Bar-M Canyon study area is located approximately 25 miles south of Flagstaff. The Bar-M watershed is part of the Mogollon Rim Plateau. It is a gently rolling landscape, dissected by many steep canyons. Elevations range from 6360-7710 feet. Our plots were located between 6800 and 7200 feet. The bedrock underlying the area consists of igneous rocks of volcanic origin. The soils, developed on basalt and cinders, are mostly silty clays and silty clay loams less than 2.6 feet deep (Brown et al. 1974).

The average annual precipitation for the area is 25.0 inches. There are two major precipitation seasons. Sixty-four percent of the precipitation falls during the winter–October through April. Thirty-two percent falls during the summer–particularly July and August (Brown et al. 1974).

The North Kaibab study area is located on the Kaibab Plateau of north central Arizona, approximately 100-120 miles north of Flagstaff. Like Bar-M canyon it is a gently rolling landscape, dissected by steep canyons. The elevation of our plots ranged from 6800-7800 feet. The bedrock underlying the area consists primarily of Kaibab limestone. The soils, developed from limestone, are mostly sandy and gravelly loams and loams. Average annual precipitation for the North Kaibab ponderosa type is 22 inches (Brewer et al. 1991). Predominant vegetation composition of the study areas is described in Table 2.

Field Procedures

On both sites we used a stratified systematic sampling procedure. The areas were stratified by soil type and topography, using the U.S. Forest Service Terrestrial Ecosystem (TE) Survey (USDA Forest Service 1987; Brewer et al. 1991). At Bar-M, map unit #582 was the most common soil-slope-vegetation unit (Typic Argiboroll and Mollic Eutroboralf; Low Sun Cold, with ponderosa pine and Gambel oak as the dominant trees, 0-15% slope). Within the North Kaibab study area, map units #293 and #294 were the most common (Mollic Eutroboralf; Low Sun Cold, with ponderosa pine and Gambel oak as the dominant trees, 0-15% slope (#293) the most common and 14-40% slope (#294) also represented).

At the Bar-M study area, seventy 0.62 acre (one-quarter hectare) plots were systematically located within map unit #582. The large plot size was chosen to incorporate the patchy nature of ponderosa pine old-growth and for spatial analysis in the future. Sixty-two of these plots were labeled extensive; only the presettlement trees were sampled on these plots. All presettlement trees were stem-mapped (exact x,y location recorded on the plot). In addition to location, presettlement tree species, dbh, condition

TABLE 2. Predominant vegetation composition of the ponderosa pine/bunchgrass ecosystems of the Kaibab Plateau and Bar-M study areas.

Common Name	Scientific Name	Growth Form
Ponderosa pine	*Pinus ponderosa*	Tree
Gambel oak	*Quercus gambelii*	Tree
Juniper	*Juniperus* spp.	Tree
Quaking aspen	*Populus treniuloides*	Tree
Douglas-fir	*Pseudotsuga menziesii*	Tree
Spruce	*Picea* spp.	Tree
Fir	*Abies* spp.	Tree
New Mexican locust	*Robinia neomexicana*	Shrub
Gambel oak sprouts	*Quercus gambelii*	Shrub
Buckbrush	*Ceonothus Jendlerii*	Shrub
Oregon grape	*Berberis repens*	Shrub
Showy aster	*Aster commutatus*	Forb
Spreading fleabane	*Erigeron divergens*	Forb
Showy goldeneye	*Viguiera multiflora*	Forb
Western ragweed	*Ambrosia psilostachya*	Forb
Snakeweed	*Gutierrezia* spp.	Forb
Lupine	*Lupinus* spp.	Forb
Mutton bluegrass	*Poa fendleriana*	Grass
Pine dropseed	*Blepharoneuron tricholepis*	Grass
Black dropseed	*Sporobolus interruptus*	Grass
Blue grama	*Bouteloua gracilis*	Grass
Bottlebrvsh squirreltail	*Sitanion hysterix*	Grass
Long-tongue mutton bluegrass	*Poa longiligula*	Grass
Arizona fescue	*Festuca arizonica*	Grass

(live, snags, stumps, and down), and density (number per acre and basal area) were recorded.

Eight of these seventy 0.62 acre plots were sampled more intensively at Bar-M. Information on all trees was gathered on these plots. In addition to location, the species (live, snags, stumps, and down), size class (e.g., seedling, sapling, etc.), dbh, and density were also recorded. A ten percent sample of all trees less than 14.6 and greater than 3.9 inches dbh was selected to determine an age distribution and approximate date of post-settlement tree establishment. All trees greater than or equal to 14.6 inches were aged as was any pine tree with yellow bark. Our logic for these dbh and bark criteria was similar to that of White (1985) who determined statistically that the majority of presettlement ponderosa pine at a similar site would be > 14.6 in., and that those which were not would have yellow bark. This latter criterion is based on the observation that, in the Southwest ponderosa pine, bark changes color from "black" to "yellow" as the tree ages (Pearson 1950).

At the North Kaibab study site we made our plot locations compatible with an earlier inventory (Lang and Stewart 1910). Our sample plots were located in the center of systematically selected quarter sections. Forty-six 0.62 acre (0.25 ha) plots were located in TE map units #293 and #294. Presettlement trees were sampled on all plots, while additional information on postsettlement trees was gathered on 36 of these plots. Data collected and sampling techniques used were the same as described above for the Bar-M study site. The results for only 16 intensive plots (data on all trees) from the North Kaibab are presented in this paper because we are still analyzing the remainder.

The tree rings of all presettlement trees from both study sites were counted and measured to determine total tree age and to determine the diameter of each tree at a point before Euro-American settlement (1867-Bar M; 1881-North Kaibab), and to determine average annual growth since settlement. If the tree was a stump, snag, or down material then the year of death was estimated (Thomas 1979; Maser et al. 1979; Cunningham et al. 1980; Rogers et al. 1984), and the presettlement diameter calculated by regression.

Simulation Models

To understand how forest structure and patterns and resource conditions changed over time we linked the spatial data from the stem-mapped intensive plots (8 from Bar-M and 16 from North Kaibab) to the ECOSIM multiresource forest growth and yield simulation model (Rogers et al. 1984). The tree growth and yield model used in ECOSIM is based on the FREP/STEMS model (Belcher et al. 1985; Brand 1981) calibrated with continuous forest inventory data from Arizona and New Mexico. The water yield model is based on the "Baker-Kovner" model (Brown et al. 1974) in which streamflow is a function of winter precipitation, aspect, slope, and tree density. For herbage production, a modification of Clary's (1978) model was used, where herbage is a function of annual precipitation, tree density, and range site class. Forest floor accumulation is estimated as the difference between litterfall (calculated from tree density and canopy biomass) and decomposition, using Fogel and Cromack's (1977) estimates for ponderosa pine decomposition rates. Near-view scenic beauty is estimated using equations developed by Daniel and Boster (1976) and Schroeder and Daniel (1981). This index of scenic beauty is calculated as a function of number of large (> 16 inch dbh) trees, number of mid-sized trees (5-16 inches dbh), amount of logging slash, amount of herbage, and amount of shrubs. For more detail on the simulation techniques the reader is referred to Rogers et al. (1984).

Each simulation run consisted of entering the dbh of all presettlement trees (live trees, snags, downed trees and stumps) by diameter class as stand conditions in 1867 for Bar-M and 1881 for the North Kaibab. Settlement of the Flagstaff area preceded that of the North Kaibab area, thus grazing and hence fire exclusion began earlier at the Bar-M study area. This paper is designed to examine the inherent properties of ponderosa pine population irruptions when fire is excluded; therefore we did not simulate historical tree harvesting in this analysis. This stands in contrast to our analytical procedure which included historical harvest at Bar-M published elsewhere (Covington and Moore 1994).

After initialization to presettlement conditions, we entered trees into the stand at appropriate intervals in simulated time, based on the regeneration events inferred from the age distribution of the postsettlement trees. The output from these computer runs was a series of tables (from 1867 or 1881 through 2021 or 2027) which quantitatively estimate the changes in forest density in southwestern ponderosa pine since Euro-American settlement. This information on forest density was used to run fuel loading, herbage, water yield, and esthetics models (Rogers et al. 1984). We used the model results to draw inferences about temporal changes in multiresource conditions since European settlement and to forecast future trends.

Results and Discussion

Tree density at both study areas has increased greatly since the late 1800s (Table 3). Presettlement tree density was higher by a factor of two on the North Kaibab study area than at the Bar-M study area. Postsettlement tree density was much greater at Bar-M than at North Kaibab. However, because of the small sample size (8), the postsettlement tree densities at Bar-M should be viewed with caution. Our estimates of presettlement tree densities are consistent with estimates from other sources (Table 4).

The results from the simulation model analysis are presented in Figures 2 through 5. We initialized the simulation in 1867 for Bar-M (although we begin plots at 1887 for chronological comparability with NKRD) and in 1881 for North Kaibab as follows:

1. Tree density–we entered the trees by dbh and species present in 1867 at Bar-M and 1881 at North Kaibab.
2. Site index = 75, the site index for Terrestrial Ecosystem Survey map units #582, #293, and #294.
3. Soil rating factor = 9 for Bar-M and 12 for North Kaibab, based on the forage production potential for map units #582, #293, and #294. Soil rating factor is an index varying from 0 for poor range sites to 12 for the best.
4. Fuel loading = 0.1 t/ac of fermentation + humus layers and 0.1 t/ac of litter layer of the forest floor. This is based on the fuel loading data from Covington and Sackett's (1986) 2-year interval prescribed burning plots in ponderosa pine with the assumption that tree canopy covered approximately 20% of the surface of the land (Covington and Sackett 1986; White 1985).
5. For Bar-M, average annual precipitation = 25 inches; average annual winter precipitation = 16.9 inches. This is the 22 year average for the ponderosa pine watersheds in the Beaver Creek drainage (Campbell and Ryan 1982). Precipitation was held constant throughout the simulation. For North Kaibab, average annual precipitation = 22 inches; average annual winter precipitation = 12.1 inches, based on climatic data reported in the Terrestrial Ecosystem Survey (Brewer et al. 1991) for north Kaibab map units #293 and #294.

To simulate the establishment of postsettlement trees we entered seedling density corresponding with the number of postsettlement trees for each species. Seedlings were entered beginning at the point in simulated time which represented the earliest establishment data observed on the plot. Similarly, the last date of seedling establishment was used as the last seedling establishment date in simulated time.

The results of the simulation model provide an estimate of how forest conditions have changed since Euro-American settlement in Arizona ponderosa pine type and how these trends might continue into the future (Figures 2-5). Increases in tree density through 1990 are estimated to have caused substantial declines in average herbage production (decreases of over 1,000 lbs/ac at Bar-M and 350 lbs/ac on the North Kaibab). This decreased herbage in conjunction with increased small diameter tree density has caused a striking decline in near-view scenic beauty (Covington and Moore 1994a). At the same time, forest floor and fuel loading is estimated to have increased from less than 1 t/ac before settlement to an average of

TABLE 3. Changes in the tree density since Euro-American settlement for two ponderosa pine study sites. Data are means ($\bar{x}$) based on varying sample sizes (n).

Study Area	Presettlement[1]		Current[1]	
	------ trees/acre ------			
	$\bar{x}$	(n)	$\bar{x}$	(n)
North Kaibab[2] (limestone)	55.9	(36)	276.3	(30)
Bar-M Canyon[2] (volcanic)	22.8	(70)	851.0	(8)

[1]includes stumps, snags, and down trees

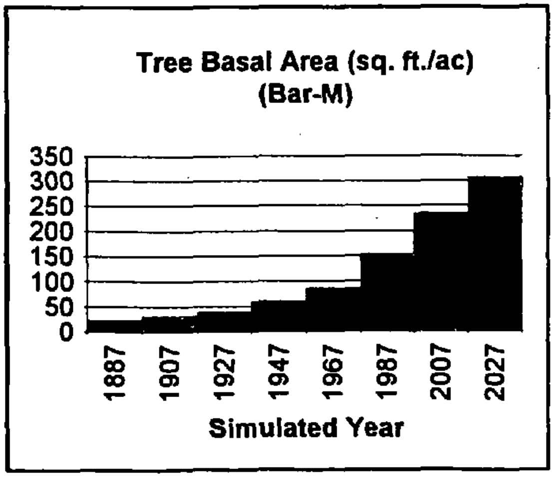
[2]TE map units #293 and #294 NKRD; TE map unit #582 Bar-M; North Kaibab sampling and data analysis are still in progress

TABLE 4. Density of southwestern ponderosa pine presettlement or yellow pine trees reported in the literature and in this study.

Location	Trees/acre
Specific studies in the Southwest:	
Ft. Valley, Coconino N.F.[1]	15
Bar-M, Coconino N.F.[2]	23
North Kaibab R.D., Kaibab N.F.[3]	56
North Kaibab R.D., Kaibab N.F.[4]	40-45
White Mountains, Apache-Sitgreaves N.F.[5]	35-45
Southern Utah, Zion N.P.[6]	22.7
USFS Bulletin 101:[7]	
(some of the heavily stocked yellow pine stands)	
Coconino N.F.	27
Kaibab (Tusayan) N.F.	35
Carson N.F.	26-47

[1]White, 1985; [2]Covington and Moore, 1994a; [3]Covington and Moore, 1994b; [4]Rasmussen, 1941; [5]Cooper, 1960; [6]Madany and West, 1983; [7]Woolsey, 1911

FIGURE 2. Tree basal area (ft^2/ac) since European settlement in the Arizona ponderosa pine type at Bar-M and at the North Kaibab Ranger District (NKRD).



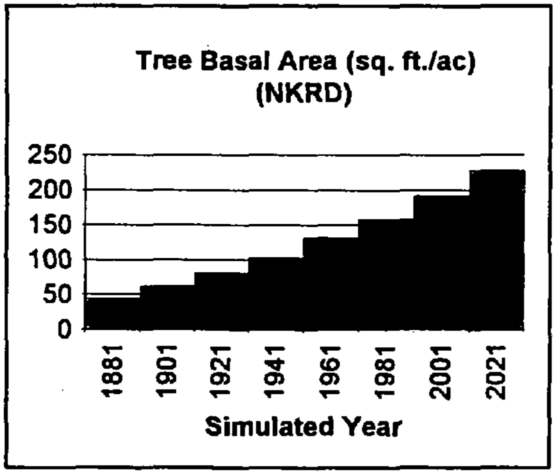


FIGURE 3. Crown closure (%) since European settlement in the Arizona ponderosa pine type at Bar-M and at the North Kaibab Ranger District (NKRD).

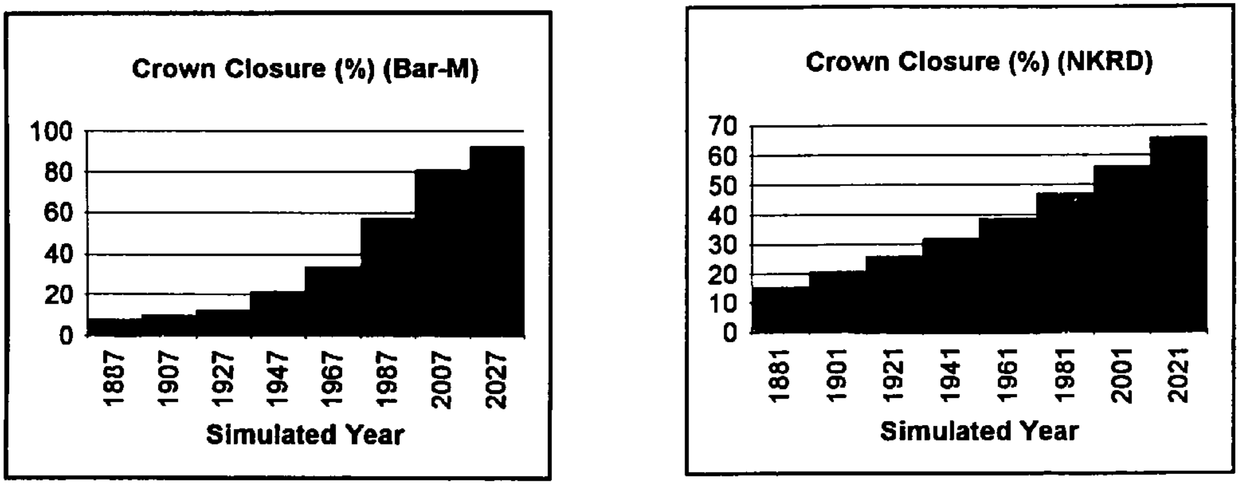


FIGURE 4. Fuel loading (t/ac) since European settlement in the Arizona ponderosa pine type at Bar-M and at the North Kaibab Ranger District (NKRD).

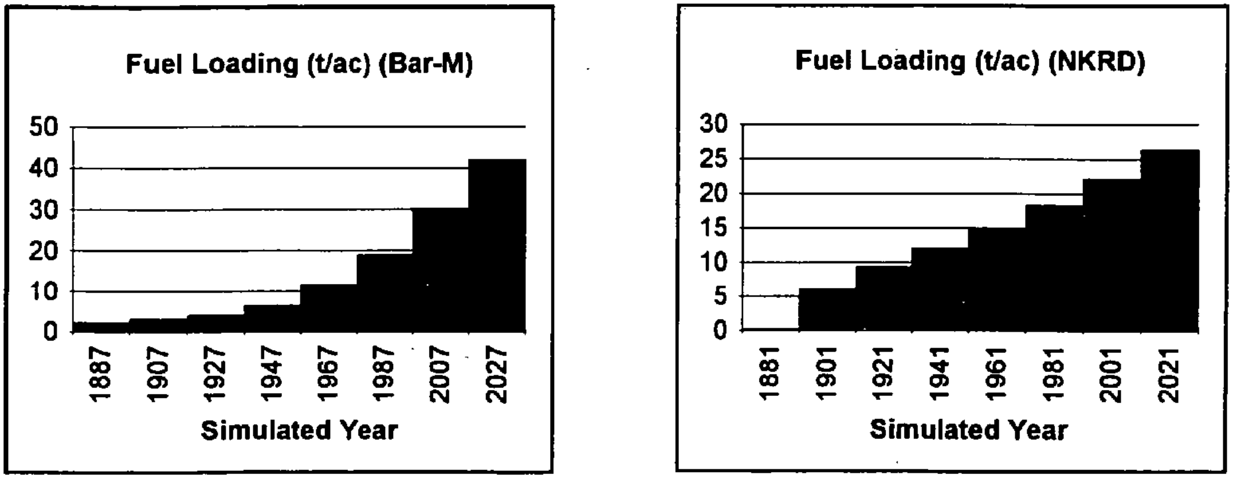
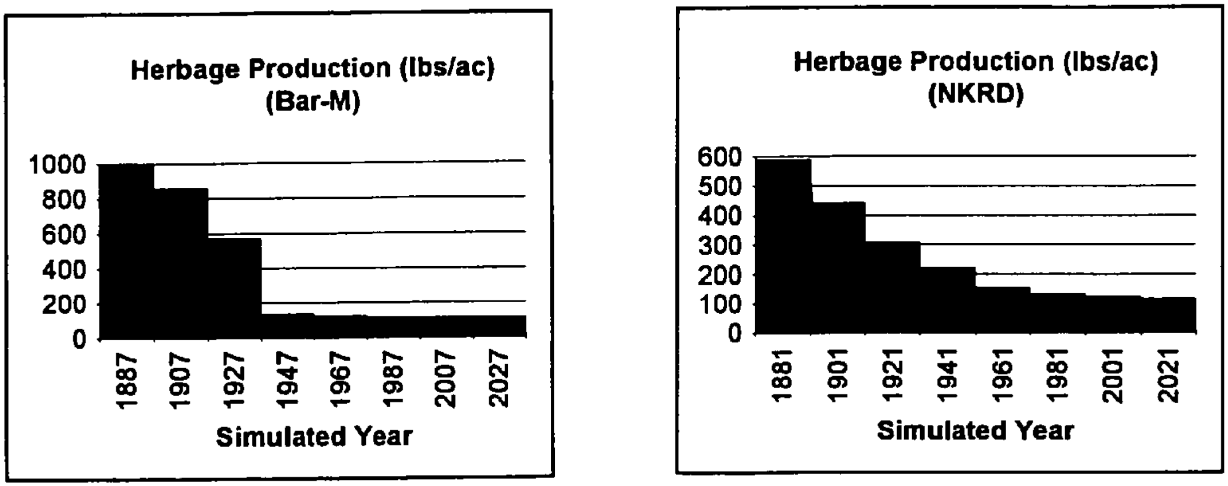


FIGURE 5. Herbage production (lbs/ac) since European settlement in the Arizona ponderosa pine type at Bar-M and at the North Kaibab Ranger District (NKRD).



over 20 t/ac at both study areas. Vertical diversity, fuel ladder continuity (as estimated by diameter distribution of trees), and crown closure have also increased substantially (Figures 2-5).

To estimate changes in wildlife habitat characteristics since 1867, we used the simulation output for tree density by diameter class to classify the simulated stand into vegetation structural stages (Thomas 1979) and then used the Forest Service Southwestern Region's forest planning wildlife report (Byford et al. 1984) to determine the change in relative habitat value from 1867 to the present. This analysis indicates that there has been a shift away from the grass-forb structural stage common in the late 1800's to a seedling dominated structural stage after the turn of the century, and then to mature timber and finally old-growth trees growing over dense sapling and poles from the 1960s on.

These changes in vegetation since settlement indicate a shift in foraging habitat from one favoring grassland/savannah species (e.g., pronghorn antelope, grasshoppers, bluebirds, and turkeys) to one favoring species feeding in dense forests (Abert squirrel, porcupine, bark beetle, pygmy nuthatches, and perhaps Mexican spotted owl). In sum, the shift in tree density seems to have favored species dependent on closed forest conditions at the expense of those which require some portion of their habitat in grass/forb or savannah. Gruell et al. (1982) noted similar changes in wildlife habitats for ponderosa pine/Douglas-fir forests in western Montana.

Under the assumption that no substantial tree mortality occurs between 1987 and 2027, many of these trends are predicted to continue (Figures 2-5). Undoubtedly, the increased tree density is a key factor contributing to the increased occurrence of large crown fires in southwestern ponderosa pine (Barrows 1978; Swetnam 1990). Fire simulation studies of ponderosa pine and related forest types (van Wagtendonk 1985; Keane et al. 1990) are consistent with these conclusions.

Furthermore, increased density is responsible for decreased growth rates of presettlement trees (Sutherland 1983), and hence decreased vigor (Waring 1983), which increases susceptibility to bark beetle attack (Sartwell 1971; Sartwell and Stevens 1975) and other agents of mortality. Numerous studies have demonstrated that the mortality of ponderosa pine increases with both diameter and stand density (McTague 1990). A trend toward increasing rates of mortality, especially of the largest and oldest presettlement trees, is supported by an analysis of the Pearson Natural Area data (Covington and Moore 1994a).

Thus, the increase in tree density following Euro-American settlement has resulted, on the one hand, in a major increase in forest canopy cover and vertical diversity within the tree canopies, and on the other hand striking decreases in herbage production. Our results are consistent with studies in Interior (Gruell et al. 1982; Keane et al. 1990) and California (van Wagtendonk 1985; Laudenslayer et al. 1989) ponderosa pine forests which describe changes in tree density, fuel loads, wildlife habitat, and esthetics after fire exclusion. Of particular concern is the increased risk of mortality, especially for the oldest age classes of presettlement trees, from crown fire, bark beetles, and other agents (such as root rot), and the implications for old-growth forest management in the ponderosa pine type.

IMPLICATIONS FOR OLD-GROWTH ECOLOGY AND MANAGEMENT

Disruptions of natural disturbance regimes coupled with postsettlement anthropogenic disturbances have led to forest and woodland conditions which may bear little resemblance to natural conditions. Thus, old-growth definitions and management objectives based only on current stand structure may not be compatible with the conservation biology goal of preserving species diversity by providing for the habitats in which species have evolved (Hunter 1991). In fact, detailed analysis of tree density, size, crown closures, and down logs indicates that, before settlement, none of the plots sampled at Bar-M nor at the North Kaibab would meet all of the current minimum criteria for old-growth used in the Southwest[1] (USDA Forest Service 1992). In addition, when we compared the current conditions of the 36 intensive plots on the North Kaibab (all were site index 70-80) to these same criteria for old-growth, only three plots met the old-growth trees/acre requirements; however, in all three cases only half of the dominant trees met the 180 year requirement, and only one of these plots met the snag, while none met the down log, requirement.

Setting aside old-growth ponderosa pine stands which most closely meet current old-growth definitions may have unexpected consequences. These stands that have higher than normal canopy closures when compared to presettlement times are likely to be the most susceptible to crown fire, low tree vigor, and mortality from drought, insects, and diseases. Further deterioration from natural conditions is inevitable if restorative actions are not taken immediately.

1. Minimum structural attributes for old-growth inventory in the USDA Forest Service's Southwestern Region–for high sites (Minor site index > 55): large dominant/codominant trees–20 trees/acre ≥ 18 inches dbh; average 180 years; total tree basal area of 90 ft^2/acre; total canopy cover of 50%; dead tree component of 1 snag/acre of 14 inches dbh and 25 feet in length; 2 down logs/acre of 12 inches dbh and 15 feet in length; canopies single or multiple storied.

Thus, definitions of old-growth should take into account natural conditions before Euro-American settlement, particularly the natural fire regime and patchy nature of the forest. In managing toward natural (using presettlement conditions as the "yardstick") old-growth, it will be necessary to design and apply treatments for restoring candidate stands (e.g., thinning from below, manual fuel treatments, prescribed burning).

RESTORATION OF SOUTHWESTERN PONDEROSA PINE

To summarize, numerous lines of evidence point to striking postsettlement changes in southwestern ponderosa pine forests. A combination of livestock grazing, fire exclusion, and logging disturbances has resulted in increases in tree density, canopy closure, vertical diversity, aerial fuel continuity, and surface fuel loads. At the same time, herbaceous and shrub production have likely declined. As a consequence, the forests of today differ substantially from the natural conditions before Euro-American settlement. These changes in ecosystem structure imply changes in wildlife habitat, water relations, nutrient cycling, soils, species diversity, ecosystem health, and other resource characteristics.

Furthermore, these changes in ponderosa pine structure have led to a shift away from the natural fire regime of frequent, low intensity surface fires to high intensity crown fires. Most recently, the occurrence of larger and larger crown fires in the ponderosa pine type may indicate a further shift to a regime characterized by very large (> 10,000 acre) crown fires.

To remedy these problems and restore these forests to more natural conditions immediate action is essential. Although it is beyond the scope of this paper to recommend specific ecological restoration prescriptions, sufficient understanding (both scientific and expert knowledge) exists for developing and testing site specific ecological restoration hypotheses at levels in the landscape hierarchy from small plots to entire landscapes. For example, in southwestern ponderosa pine/bunchgrass ecosystems such as those reported in this paper, existing knowledge indicates that in dense sites with heavy forest floor accumulations, heavy fuels must be removed from the base of large, old trees (or larger replacement postsettlement trees in the absence of old-growth), periodic burning must be reintroduced, and native understory species must be reestablished.

However, simultaneous with widescale restoration treatments, it is necessary to determine postsettlement changes in ecological conditions (Bonnicksen and Stone 1985) for a broader range (e.g., a variety of soil types, topographies, and vegetation types) of forest and woodland types in the

Rocky Mountain and Southwest regions. Then a combination of an adaptive resource management approach (Walters 1986) and process-oriented simulation modeling (e.g., van Wagtendonk 1985; Keane et al. 1990; Covington and Moore 1994a) could be used to design restoration management regimes appropriate for each set of conditions. Simultaneously, small (10-20 acre) plot studies could be established to examine the ecological effects and practicality of various restoration treatment scenarios. Using such an integration of historical studies, simulation modeling, and management experiment approaches should ensure that we address our wildland management issues in a more coherent and ecologically sound manner.

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